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RESEARCH ARTICLE

Biochar co-compost improves nitrogen retention and reduces carbon emissions in a winter wheat cropping system

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Abstract

Organic amendments, such as compost and biochar, mitigate the environmental burdens associated with wasting organic resources and close nutrient loops by capturing, transforming, and resupplying nutrients to soils. While compost or biochar application to soil can enhance an agroecosystem's capacity to store carbon and produce food, there have been few field studies investigating the agroecological impacts of amending soil with biochar co-compost, produced through the composting of nitrogen-rich organic material, such as manure, with carbon-rich biochar. Here, we examine the impact of biochar co-compost on soil properties and processes by conducting a field study in which we compare the environmental and agronomic impacts associated with the amendment of either dairy manure co-composted with biochar, dairy manure compost, or biochar to soils in a winter wheat cropping system. Organic amendments were applied at equivalent C rates (8 Mg C ha^{-1}). We found that all three treatments significantly increased soil water holding capacity and total plant biomass relative to the no-amendment control. Soils amended with biochar or biochar co-compost resulted in significantly less greenhouse gas emissions than the compost or control soils. Biochar co-compost also resulted in a significant reduction in nutrient leaching relative to the application of biochar alone or compost alone. Our results suggest that biochar co-composting could optimize organic resource recycling for climate change mitigation and agricultural productivity while minimizing nutrient losses from agroecosystems.

KEYWORDS

biochar co-compost, climate change mitigation, dairy manure management, nitrogen leaching, soil greenhouse gas, soil health

INTRODUCTION

Food production nourishes the world, but it also warms the planet, emitting up to one-third of total greenhouse gas emissions (Crippa et al., 2021). While soils store up to three times as much carbon (C) as the atmosphere, the industrialization of agriculture has led to the rapid loss of C from soils (Sanderman et al., 2017). This loss of soil organic C (SOC) can compromise an agroecosystem's capacity to produce food and can transform it from a net greenhouse gas sink into a net source (Lal, 2015). Global efforts such as the 4 per mille initiative aim to combat climate change through the widespread adoption of agricultural practices that increase SOC while promoting soil health (Minasny et al., 2017).

One strategy to increase SOC involves the application of organic amendments, such as compost and biochar, to soils. In addition to offering a sustainable alternative to the landfilling and burning of organic resources, compost and biochar can enhance SOC sequestration, increase soil fertility, improve soil structure and water retention and increase crop yield (Chen et al., 2018; Diacono & Montemurro, 2010; Martínez-Blanco et al., 2013). Circular economies that are based on the transformation of local agricultural residues into compost or biochar also allow for the capture and reuse of nutrients that would have otherwise been lost from the agroecosystem, reducing the need for synthetic fertilizers.

Agricultural regions, such as California's Central Valley, produce a vast amount of organic waste in the form of woody debris and livestock manure. Current waste management practices such as biomass burning and solid manure stockpiling have severe consequences, contributing to climate change, ecosystem degradation and pollution that can harm human health (Ngo et al., 2010). While producing wood biochar and manure compost offer sustainable alternatives to burning and stockpiling, combining these practices in a process called biochar co-composting may optimize the environmental and agronomic benefits from utilizing these contrasting organic resource streams. For example, studies have found that composting nitrogen (N)-rich, dense manure with C-rich, porous biochar can substantially reduce greenhouse gases (GHGs) and nutrient losses during composting as biochar can improve compost aeration, adsorb nutrients and gases and enhance microbial activity (Agyarko-Mintah et al., 2017; Harrison et al., 2022; Sanchez-Monedero et al., 2018).

While a number of studies have investigated the biochar co-composting process, there have been very few studies that examine the use of biochar co-compost (defined here as the product of biochar co-composting) as a soil amendment (Kammann et al., 2017; Sanchez-Monedero et al., 2018; Wang et al., 2019); however, numerous Indigenous peoples have used similar charcoal-organic residue mixtures to improve the fertility of soils for millennia (Glaser et al., 2001). Some studies have shown that biochar co-compost may improve crop yield and/or soil health, but data are limited and few studies have been conducted at the field scale (Agegnehu et al., 2016a, 2017; Bass et al., 2016; Wang et al., 2019; Yuan et al., 2017). Despite a lack of data, several studies suggest that synergistic effects between biochar and compost may lead to agronomic advantages of biochar co-compost compared to other amendments (Kammann et al., 2015; Pandit et al., 2020; Schulz et al., 2013). These advantages may be due to a higher retention of initial feedstock N as biochar has been shown to reduce N losses during co-composting (Yuan et al., 2017), where the adsorption of both ammonium (NH_4^+) and nitrate (NO_3^-) on the biochar surface can form a mixed-charged organo-mineral layer through the composting process (Archanjo et al., 2017; Hagemann et al., 2017; Kammann et al., 2015; Prost et al., 2013). However, much remains unknown about how agroecosystems respond to biochar co-compost application, especially when considering both agronomic (e.g., crop yield, soil water retention, nutrient availability, leaching, and retention) and environmental (e.g., soil GHG emissions, C sequestration, and soil health) impacts relative to the application of compost or biochar on its own (Harrison et al., 2022; Wang et al., 2019; Yuan et al., 2017).

The objective of this study was to evaluate the effects of biochar, dairy manure compost, and biochar co-compost application to agricultural soils on agronomic and environmental outcomes. We hypothesized that the biochar co-compost treatment would result in the highest soil plant-available N retention, the greatest crop yield, and the lowest GHG emission due in part, to its high C content and its capacity to provide and retain plant-available nutrients (Steiner et al., 2015). Compost and biochar co-compost were produced at a local dairy farm (Harrison et al., 2022) and amendments were applied to a winter wheat field, adjacent to the dairy, that produces silage for the dairy cows. To our knowledge, this is the first study to consider the agroecological impacts of closing a nutrient and C loop between dairy cattle and feed through the production and application of biochar co-compost. This study is also the first to compare fluxes of nitrogen oxides (NO_x) from soils applied with either biochar, compost, or biochar co-compost at a field scale.

MATERIALS AND METHODS

Study site description and field experimental design

A field experiment was established in October 2021 at a conventional winter wheat (*Triticum aestivum*) cropping field at the Philip Verwey Farm at Madera, CA, USA (36.945°N, 120.378°W). The region exhibits a Mediterranean climate with a mean annual temperature of 18.2°C and a mean annual precipitation of 311 mm. Summer is hot and dry (e.g., July average temperature is 29.3°C with an average precipitation of 0 mm) while winter is relatively cool and damp (e.g., January average temperature is 8.5°C with 56 mm monthly average precipitation). Soils of the study site are classified as loamy, thermic Natric Durixeralf USDA Soil Taxonomic family (soil order: Alfisols) and is moderately well-drained. The parent material is alluvium derived from granite. Surface soil (0–30 cm) has a sandy loam texture (15% clay, 67% sand, 18% silt, determined using the hydrometer method, Ashworth et al., 2007), 27.0 $g\,kg^{-1}$ total C, and 1.4 $g\,kg^{-1}$ total N. Soil pH is 8.03 measured in 1:2 soil to DI water (v/v) suspension, and the bulk density of the soil is 1.36 $g\,cm^{-3}$.

Replicated treatment plots were established in a randomized block design ($n = 4$) at the study site. Raised beds formed during planting divided the field into sections approximately 16 m wide (Figure S1), therefore we treated each field section as an independent replicated block ($n = 4$). Four treatment plots (2 m × 12 m with 2 m buffer) were established within each replicated block (Figure S1). Treatments included unamended controls and three treatment plots with organic amendments: manure compost, biochar, and biochar manure co-compost, each applied at equivalent C rates of 8 Mg C ha^{-1}, resulting in manure compost applied at 20 Mg ha^{-1}, biochar applied at 10 Mg ha^{-1}, and biochar manure co-compost applied at 17.5 Mg ha^{-1} (all dry weight basis). Treatments were randomly assigned to plots within each replication block. All treatments were applied to the surface soil of the plot and incorporated to approximately 15 cm depth with a rake and pitchfork 1 week prior to winter wheat seeding. We also raked control plots to ensure consistency in soil disturbance. Our study site received no synthetic fertilizer input and were irrigated twice by the landowner over the course of the experiment (i.e., 2 weeks following seeding, and 1 week following our mid-season sampling). Monthly precipitation and average temperature at the study site for the duration of the field experiment are summarized in Table S1. All plots received the same management following the landowner's practices. The winter wheat was harvested at the booting stage and the experiment lasted for approximately 5 months.

Biochar, manure compost, and biochar manure co-compost

Biochar used in the field experiment was Rogue biochar from Oregon Biochar Solutions (White City). The feedstock of biochar consisted of approximately 85% Douglas fir (*Pseudotsuga menziesii L.*) and ponderosa pine (*Pinus ponderosa L.*) wood waste mixture, 14%–15% almond and walnut tree pruning, and less than 1% of nutshells. The maximum pyrolysis temperature was reported to be 900°C (personal communication). Information on the production and characteristics of biochar can be found at chardirect.com. Biochar particles had diameters ranging from 3 mm to 1 cm. Compost and biochar co-compost were both prepared on-site in compost windrow piles at the Philip Verwey Dairy during late summer 2021. Each pile was trapezoidal in shape and approximately 30 m in length, 3 m in width, and 1 m in height. The compost pile consisted of approximately 15.34 t fresh solid dairy manure and 1.32 t orchard clipping residues. Biochar co-compost pile consisted of 15.35 t fresh solid dairy manure, 1.32 t orchard clipping residues, and 1.0 t biochar (Harrison et al., 2022). We used a 6% (w/w) biochar application rate as low rates of biochar application (3%–9%) have been shown to have beneficial effects on the composting process such as improved nutrient retention and reduced greenhouse gas emissions while higher biochar application rates may not be cost-effective for farmers (Hua et al., 2009; Sanchez-Monedero et al., 2018; Steiner et al., 2015; Wang et al., 2013). Dairy manure in both piles was directly sourced from the dairy farm on-site. Both compost piles were turned weekly. Biochar was characterized with proximate analysis for volatile matter content, fixed C content, and ash content. The BET surface area and porosity characteristics of biochar were quantified using a TriStar II Plus gas adsorption analyzer (Micromeritics Dr. Norcross, GA) following (Bardestani et al. 2019). The surface roughness of biochar was quantified using a ZEISS Gemini 500 FE-SEM (Zeiss) following Zheng et al. (2021). Characteristics of biochar, compost, and biochar co-compost are summarized in Table S2.

Soil sampling, analyses, and nutrient leaching at the rooting zone

Three soil samples were collected and composited from each treatment plot at five depth intervals (0–10, 10–30, 30–50, 50–75, and 75–100 cm) using 5.7 cm-diameter soil augers at the early- (October 2021), mid- (January 2022), and end-growing season (March 2022). Bulk density was determined using a quantitative corer (7 cm diameter). Soil infiltration tests were carried out in the field using the single ring infiltrometer method (Chowdary et al., 2006). Surface soil samples were homogenized and visible root tissues were removed from soil samples before being analyzed for a series of soil physicochemical and biochemical variables. Soil pH and electrical conductivity (EC) were determined on fresh soil (1:2 v/v soil-to-DI water) on a pH/EC meter (Mettler Toledo SevenCompact, S220, S230). Fresh soil samples were weighed, shaken in 2 M KCl for 1 h, filtered through Whatman 1 filter papers, and the extractants were analyzed for extractable NO_3^--N and NH_4^+-N by microplate-colorimetric techniques using the vanadium-chloride method and salicylate-nitroprusside method, respectively (Mulvaney et al., 1996). Soil net N mineralization rates were determined using the 28-day aerobic incubation method described in Hart et al. (1994), and were calculated by subtracting the initial extractable N (day 0) from that determined at the end of the incubation (day 28). Nitrification potential was determined on fresh soils using the aerated slurry method described in Hart et al. (1994). Soil bioavailable phosphorus (P) status was determined using the biologically based P method described in DeLuca et al. (2015) to assess a suite of four plant P acquisition strategies to evaluate P bioavailability in dynamic soil systems. Briefly, 0.01 M $CaCl_2$, 10 mM citric acid, 0.2 enzyme unit ml-1 phosphatase enzyme (derived from wheat germs), and 1 M HCl were used as extractants to emulate free soluble P, active inorganic P (weakly sorbed or bounded in inorganic precipitates), active or labile organic P (readily attacked by phosphatase enzymes), and moderately stable inorganic P (present in P-precipitates). Extracts were analyzed for orthophosphate using the Malachite green method (Ohno & Zibilske, 1991). Soil microbial biomass C was determined by the chloroform fumigation extraction method (Vance et al., 1987) and the fumigated and non-fumigated K_2SO_4 extracts were analyzed on a total organic C (TOC) analyzer (TOC/TN 5050 Analyzer, Shimadzu Scientific Instruments). Water holding capacity was determined by gravimetry (Loveday, 1974). Soil wet aggregate stability was determined by wet-sieving air-dried soils using nested sieves submerged in water following Kemper and Rosenau (2018) and was reported as the mean weight diameter following Rath et al. (2022) and van Bavel (1950). Oven dried (65°C) samples were ground and analyzed for total C and N on an elemental analyzer (Costech ECS 4010 CHNS-O). We analyzed the extractable NO_3^--N and NH_4^+-N on soil samples collected down to 1 m at both mid- and end-growing seasons in order to reveal the spatial distribution of N availability over time. All other soil physicochemical and biochemical analyses were conducted on soils collected at the end-growing season at top 30 cm only.

At the beginning of the field experiment, one suction lysimeter per plot was installed at 50 cm depth within each treatment plot to measure soil leachate over time. Lysimeters were vacuum pressurized to approximately 70 PSI for 24 h before individual leachate sample collection (Rath et al., 2022). Approximately 15 ml leachate sample was collected from each treatment plot once per month and all samples were analyzed for inorganic N and orthophosphate concentrations following methods described above.

Soil gas flux measurements

Soil GHG (CO_2, N_2O, and CH_4) fluxes were measured in the field every other day during the first week following treatment application, every week in the first month (October–November 2021), and every other week from the second month (November 2021) to the end of the experiment (March 2022). At the beginning of the field experiment, we randomly picked two locations within each treatment plots for collar installation and gas measurements in order to capture the spatial variation of the treatment plot. We used a cavity ring-down laser spectrometer (Picarro G2508, Picarro Inc.) connected to a closed system static chamber (made from polyvinyl chloride and 26 cm diameter by 13 cm tall) for GHG measurements. Briefly, collars (made from polyvinyl chloride and 25.5 cm diameter by 15 cm tall) were inserted 3 cm into the soil surface and allowed to sit for 1 h before measurement. Emergent vegetation was carefully removed from collars prior to gas measurements. The chamber lid is fitted with two 1/4"tube fittings (Swagelok, Solon, OH) each connected to an inlet or outlet 1/4" tube with sample air flowing at a rate of 275 ml min^{-1}. After taking a measurement, gas concentrations were allowed to return to ambient concentrations before the next measurement. Air temperature, soil temperature and soil volumetric water content at top 10 cm were measured along with individual soil GHG measurement. Gas fluxes (nmol m^{-2} s^{-1}) were calculated in the Picarro Soil Flux Processor program using the exponential model developed by Hutchinson and Mosier (1981) to account for nonlinear changes in headspace concentration. The average gas flux from the two sampling locations was considered representative of the treatment plot and served as our analysis unit. We calculated the global warming potential on a 100-year frame by converting CH_4 and N_2O to CO_2 equivalents using 27.2 and 273 as the conversion factors, respectively, following the IPCC sixth assessment report (Pörtner et al., 2022).

Soil NO and NO_2 fluxes were measured on the same days as other gases and were measured using a static chamber system connected to a chemiluminescent NO_x analyzer (Serinus 40 Oxides of Nitrogen Analyzer, Acoem). Polyvinyl chloride soil collars (25.5 cm in diameter and 15 cm tall) were inserted 3 cm into soil at least 30 min before each measurement. Emergent vegetation was carefully removed from collars prior to gas measurements. For each measurement, a polyvinyl chloride chamber lid was placed over the soil collar creating a 26 cm diameter by 13 cm tall chamber with a total volume of 12,271.9 cm^3. The chamber was fitted with a small mixing fan and a vent that allows for makeup ambient air to enter the chamber. Air was drawn from the chamber to the analyzer at a rate of 0.6 $L\,min^{-1}$ through 3 m of FEP tubing. Concentrations of NO and NO_2 were recorded for 5 minutes for each measurement, and the rate of change in concentration was determined through linear regression over at least 150 seconds of data (Oikawa et al., 2015). Soil NO and NO_2 fluxes were calculated using the change in gas concentration, the chamber dimensions, and air temperature (Homyak et al., 2016). Soil NO_x flux was determined by adding the NO and NO_2 fluxes together. The flux rate was determined as $F = dC/dt \times VN/ART$, where F is the flux rate ($ng\,N\,m^{-2}\,s^{-1}$), dC/dt the rate of change of gas concentration (ppbv N s^{-1}), V the chamber volume (L), N the molar mass of N ($g\,mol^{-1}$), A the chamber area (m^2), R the gas constant ($L\,atm\,mol^{-1}\,K^{-1}$), and T the chamber air temperature (K).

Plant biomass and nutrient concentration

Aboveground plant biomass samples and belowground root biomass were collected from all treatment plots at the end-growing season. Three 25 cm × 25 cm quadrats were randomly located in each plot. Within each quadrat, all aboveground plant tissues were clipped and one in-situ root core (10 cm diameter × 5 cm depth) was collected and transported on ice. Aboveground plant biomass samples were oven-dried for 72 h at 65°C and weighed. Root cores were gently washed using DI water, and root samples were oven-dried for 72 h at 65°C and weighed. Aboveground plant biomass was calculated using the quadrat area and dry mass, and belowground root biomass was calculated using the root core area and dry root mass. Biomass data from three quadrats or cores were averaged to give one biomass per treatment plot. Plant total biomass (aboveground shoot biomass and belowground root biomass) was reported as kg dry weight m^{-2}. Oven-dried aboveground plant samples were ball milled and analyzed for total N on an elemental analyzer (Costech ECS 4010 CHNS-O) at the Stable Isotope Ecosystem Laboratory at the University of California, Merced; and macro- and micronutrients (e.g., S, P, K, Mg, Ca, Zn, Mn, Fe, and Cu) on an inductively coupled plasma optical emission spectrometry (ICP-OES) following a dry-ash and acid digestion procedure (Górecka et al., 2006; Munter et al., 1984) at the Environmental Analytical Laboratory at the University of California, Merced.

Statistical analyses

All data were tested for homogeneity of variance and normality of residuals before analyses, and were log transformed when necessary. Cumulative gas emissions were estimated by calculating the area under the weekly or biweekly emission curves using the function auc() in package "flux" in R (Jurasinski et al., 2012; Leytem et al., 2011). Analysis of variance (ANOVA) and Tukey's post-hoc tests were carried out on individual soil metrics, monthly nutrient leaching status below the rooting zone, gas emissions, and plant nutrient concentrations to examine the significance of treatment effect at $p = 0.05$. Pearson correlation tests were conducted on selected variables relevant to the relationships between greenhouse gas emissions and soil characteristics over the season. When necessary, we also used mixed linear regression (MLR) models to determine the dominant drivers controlling plant total biomass or cumulative soil gas emissions over one growing season. All statistical analyses and data visualizations were performed using R (R Core Team, 2020). Data visualizations were adopted from R packages described in Sievert (2020) and Wickham (2016).

RESULTS

Soil physicochemical responses and nutrient leaching below the rooting zone

We found that the addition of compost significantly increased surface soil total N, NH_4^+-N, net N mineralization, and microbial biomass C compared to the unamended control in a conventional winter wheat cropping field (Table 1 and Figure 1c). Biochar increased soil water holding capacity and infiltration rates, and soils amended with co-compost had the highest water holding capacity and infiltration rate among all treatments (Table 1). Biochar and co-compost application significantly increased soil EC, total C content, microbial biomass C, net N mineralization rate, and nitrification potential compared to control or compost application (Table 1). Compared to compost alone or biochar alone, biochar co-compost application resulted in the highest soil total N, net N mineralization, nitrification potential, NO_3^-- N, microbial biomass C, soluble inorganic P, active inorganic P, and labile organic P availability (Table 1 and Figure 1d). Soils without any organic amendment were alkaline in nature and had a pH of 8.1, but all three organic amendments reduced and buffered soil pH (Table 1). Organic amendments did not significantly alter soil aggregate stability over one growing season.

TABLE 1 Soil physicochemical properties in response to compost, biochar, and biochar co-compost 5 months following treatment application in a field experiment in Madera, California, USA.

	Control	Compost	Biochar	Co-compost
Soil water holding capacity	0.34 ± 0.01 d	0.38 ± 0.01 c	0.45 ± 0.01 b	0.48 ± 0.01 a
Bulk density ($g\,cm^{-3}$)	1.36 ± 0.06	1.38 ± 0.03	1.45 ± 0.06	1.37 ± 0.05
Mean weight diameter (mm)	0.447 ± 0.071	0.503 ± 0.054	0.509 ± 0.079	0.476 ± 0.061
Infiltration rate ($cm\,s^{-1}$)	0.04 ± 0.01 c	0.20 ± 0.08 b	0.25 ± 0.07 ab	0.38 ± 0.05 a
pH	8.10 ± 0.04 a	7.95 ± 0.03 b	7.85 ± 0.10 b	7.77 ± 0.08 b
EC (uS/cm)	147.9 ± 16.9 b	136.2 ± 13.6 b	213.0 ± 18.0 a	179.2 ± 15.8 a
Total C ($g\,kg^{-1}$)	27.0 ± 5.4 b	24.8 ± 2.6 b	37.8 ± 11.0 a	32.0 ± 5.1 a
Total N ($g\,kg^{-1}$)	1.4 ± 0.2 c	1.9 ± 0.1 b	2.0 ± 0.4 ab	2.5 ± 0.3 a
NH_4^+–N ($mg\,kg^{-1}$)	5.92 ± 1.38 b	9.73 ± 1.45 a	4.30 ± 0.86 b	4.92 ± 1.20 b
NO_3^--N ($mg\,kg^{-1}$)	0.53 ± 0.04 b	0.65 ± 0.01 ab	0.67 ± 0.03 ab	0.74 ± 0.02 a
Microbial biomass C ($g\,kg^{-1}$)	0.354 ± 0.053 c	0.392 ± 0.104 b	0.318 ± 0.060 b	0.542 ± 0.096 a
Net N mineralization ($mg\,kg^{-1}\,day^{-1}$)	0.106 ± 0.012 c	0.140 ± 0.013 b	0.142 ± 0.025 b	0.192 ± 0.008 a
Nitrification potential ($mg\,g^{-1}\,h^{-1}$)	3.20 ± 0.41 b	2.98 ± 0.23 b	4.33 ± 0.10 a	4.64 ± 0.11 a
Soluble inorganic P ($mg\,kg^{-1}$)	1.05 ± 0.13 b	1.30 ± 0.19 b	1.33 ± 0.12 b	3.45 ± 1.20 a
Active inorganic P ($mg\,kg^{-1}$)	41.6 ± 6.41 b	46.69 ± 2.96 b	41.94 ± 4.44 b	70.67 ± 13.70 a
Labile organic P ($mg\,kg^{-1}$)	19.13 ± 4.12 b	28.33 ± 0.61 b	29.86 ± 6.25 ab	48.28 ± 9.39 a
Moderately stable inorganic P ($mg\,kg^{-1}$)	136.0 ± 2.34	147.7 ± 4.48	141.3 ± 2.88	148.5 ± 5.37

Note: Data are presented as mean ± 1 × SE ($n = 4$). Numbers with the same letter are not significantly different at $p = 0.05$ and no letter following the numbers indicate no significant differences among treatments at $p = 0.05$.

At mid-growing season (3 months following organic amendment application), soils without organic amendment exhibited some amount of NH_4^+-N leaching loss below the rooting depth (over 50 cm) (Figures 1a and 2a). In contrast, biochar co-compost significantly increased surface NH_4^+-N content while reducing the NH_4^+-N downward translocation to deeper soil depth (Figures 1a and 2a). Similarly, biochar co-compost resulted in relatively less NO_3^--N leaching loss below the rooting depth compared to control or other organic amendments (Figures 1b and 2b). The reduction of NH_4^+-N leaching below the rooting depth by biochar co-compost application lasted until the end-growing season (Figures 1c and 2a). In the surface soil of the biochar co-compost treatment, the NH_4^+-N content was not as high as that in soil plots with compost alone at the end-growing season (Figure 1c). The application of compost alone resulted in some NO_3^--N leaching below the rooting zone towards the end of the growing season; in contrast, soils amended with biochar or biochar co-compost had significantly lower NO_3^--N leaching loss below the rooting zone at the end-growing season (Figures 1d and 2b). During the late season (months 3–5), soils amended with biochar co-compost also consistently exhibited less ortho-P leaching loss below the rooting zone compared to the compost treatment (Figure 2c).

Soil CO_2, CH_4, N_2O, and NO_x emissions in response to organic amendments

Compared to control and compost, biochar and biochar co-compost application significantly reduced soil cumulative CO_2 emissions by 25%–26% over one growing season (Figure 3a and Table 2). The majority of the treatment differences in CO_2 fluxes among organic amendments occurred in the first month of the field experiment. Soils amended with biochar co-compost had relatively higher CH_4 uptake compared to other organic amendments throughout the experiment; although, the cumulative CH_4 uptake at the end-growing season was not statistically significant among treatments (Figure 3b and Table 2). It is interesting to note that soils amended with biochar had consistently lower soil CH_4 uptake compared to control, compost, or biochar co-compost (Figure 3b). Cumulative N_2O emission was not significantly affected by treatment (Figure 3c and

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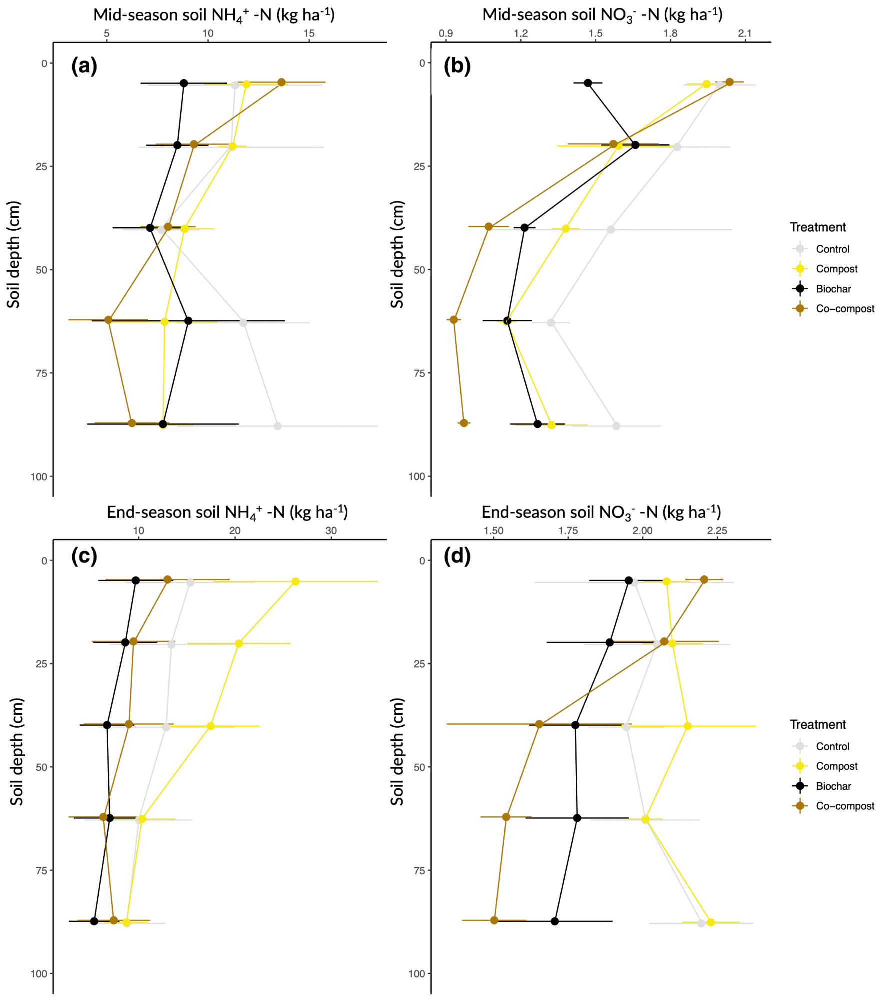

FIGURE 1 Spatial distribution of soil (a) NH_4^+ -N in mid-season (January 2022), (b) NO_3^- -N in mid-season (January 2022), (c) NH_4^+ -N in end-season (March 2022), and (d) NO_3^- -N in end-season (March 2022) (all in kg ha^{-1}) in response to compost, biochar, and biochar co-compost application in a field experiment in Madera, California, USA. Error bar represents 1× standard error.

Table 2). Similarly, differences in cumulative NO_x emissions were not significantly different across treatments (Table 2). While soils from each treatment showed a cumulative NO_x uptake due to negative NO_2 fluxes, biochar amended soils had the lowest rate of NO emission followed by biochar co-compost, compost and control. Changes in the total cumulative greenhouse gas emission in CO_2e were largely driven by changes in cumulative CO_2 emissions, and both biochar and biochar co-compost application resulted in significantly lower CO_2e (g m^{-2} or g kg^{-1} C input) compared to control or compost (Figure 5a and Table 2). We also found that soil volumetric water content was positively correlated with soil CH_4 flux (mg m^{-2} day^{-1}) and negatively correlated with soil CO_2 flux (g m^{-2} day^{-1}) throughout the season (Figure 4). Variation in cumulative CH_4 emission was found to be predominantly explained by organic amendments and their influence on soil pH and water holding capacity (Table S3). Variation in cumulative CO_2 emission was explained by organic amendments and changes in water holding capacity and microbial biomass C (Table S4).

Plant performance

Compost and biochar co-compost application significantly increased plant tissue concentrations of total P, Ca, and Fe compared to control or biochar (Table S5). Compared to control, plant aboveground biomass was significantly higher with compost application, but not with the other organic amendments (Table 2). In contrast, all three organic amendments significantly increased plant belowground biomass (Table 2). Biochar co-compost application resulted in the highest plant belowground biomass, plant total biomass, the lowest cumulative CO_2 emission (g kg^{-1} plant total biomass), and the lowest total greenhouse gas emission in CO_2e (g kg^{-1} plant total biomass) among all treatments (Table 2 and Figure 5b,c). Our multiple linear regression model suggests that the variation in plant total biomass was partially explained by organic amendment application (compost and biochar co-compost, in particular), changes in soil pH, net N mineralization, NO_3^-- N availability, microbial biomass C, soluble inorganic P, and labile organic P (Table 3).

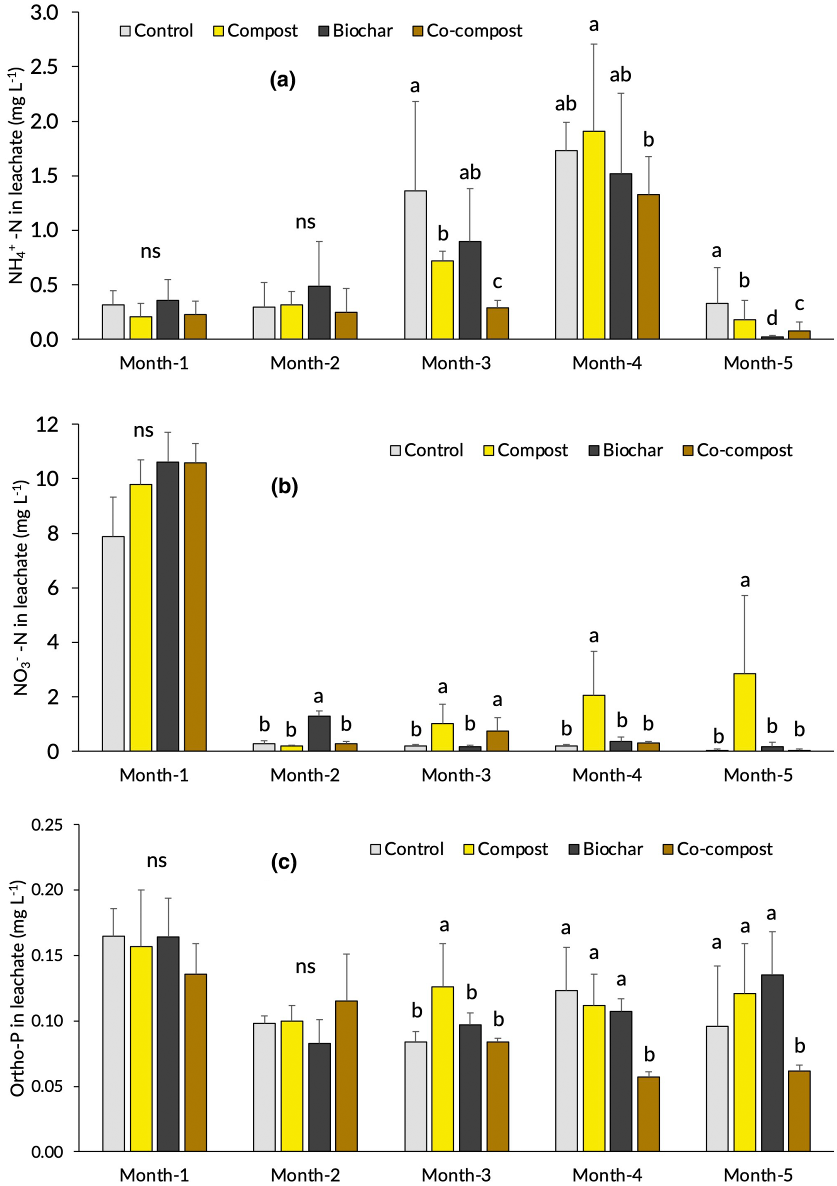

FIGURE 2 Concentrations of (a) NH_4^+ -N, (b) NO_3^- -N, and (c) ortho-P in leachate samples collected below the rooting zone over one growing season (October 2021–March 2022) in a field experiment in Madera, California, USA. Data were compared among treatments for each month using Tukey-HSD test following ANOVA. Bars with the same letter annotated are not significantly different at $p = 0.05$, ns indicates not significant at $p = 0.05$.

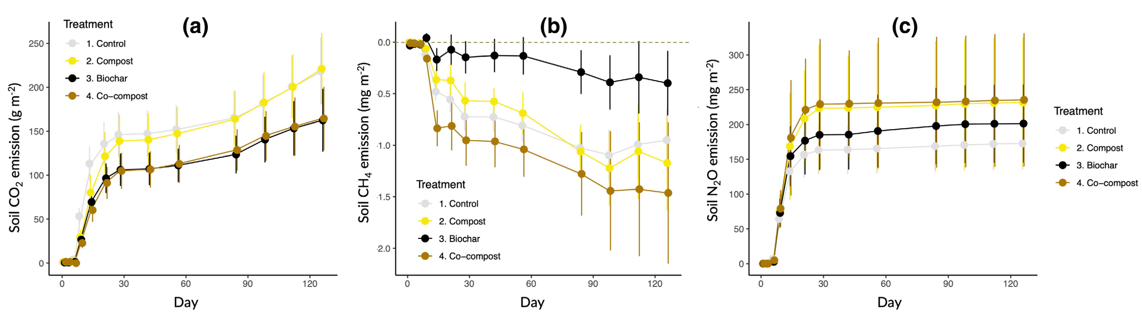

FIGURE 3 Cumulative soil (a) CO_2 emission ($g\,m^{-2}$), (b) CH_4 emission ($mg\,m^{-2}$), and (c) N_2O emission ($mg\,m^{-2}$) in response to compost, biochar, and biochar co-compost over one growing season (October 2021–March 2022) in a field experiment in Madera, California, USA. Error bar represents 1× standard error.

TABLE 2 Influence of compost, biochar, and biochar co-compost on soil greenhouse gas emission, NO_x emission, global warming potential, and plant biomass 5 months following treatment application in a field experiment in Madera, California, USA.

	Control	Compost	Biochar	Co-compost
Cumulative CO_2 emission ($g\,m^{-2}$)	218.42 ± 17.87 a	221.19 ± 31.36 a	163.00 ± 15.22 b	164.46 ± 17.86 b
Cumulative CH_4 uptake ($mg\,m^{-2}$)	0.95 ± 0.34	1.18 ± 0.37	0.40 ± 0.62	1.46 ± 0.16
Cumulative N_2O emission ($mg\,m^{-2}$)	172.72 ± 36.07	242.95 ± 113.43	201.33 ± 70.32	235.44 ± 143.14
Total greenhouse gas emission in CO_2e ($g\,m^{-2}$)	265.55 ± 26.64 a	287.48 ± 61.76 a	217.95 ± 34.26 b	228.69 ± 56.47 b
Total greenhouse gas emission in CO_2e ($g\,kg^{-1}$ C input)	-	360 ± 81.0 a	273 ± 45.0 b	286 ± 75.0 b
Cumulative NO emission (ng-N m^{-2})	817.70 ± 280.49	678.07 ± 159.23	292.97 ± 178.64	480.07 ± 100.10
Cumulative NO_2 emission (ng-N m^{-2})	−2477.34 ± 787.15	−2998.50 ± 485.18	−2236.26 ± 770.63	−2167.95 ± 482.09
Cumulative NO_x emission (ng-N m^{-2})	−1659.64 ± 938.27	−2320.43 ± 452.84	−1943.26 ± 912.66	−1687.88 ± 559.24
Plant aboveground biomass ($kg\,m^{-2}$)	0.63 ± 0.03 b	0.78 ± 0.07 a	0.62 ± 0.08 b	0.66 ± 0.03 ab
Plant belowground biomass ($kg\,m^{-2}$)	0.66 ± 0.08 c	0.94 ± 0.15 b	0.91 ± 0.13 b	1.23 ± 0.21 a
Plant total biomass ($kg\,m^{-2}$)	1.29 ± 0.08 b	1.72 ± 0.22 ab	1.53 ± 0.20 ab	1.90 ± 0.23 a
Cumulative CO_2 emission ($g\,kg^{-1}$ total biomass)	168.83 ± 6.53 a	139.15 ± 31.40 ab	115.63 ± 26.35 ab	91.13 ± 14.42 b
Cumulative CH_4 uptake ($mg\,kg^{-1}$ total biomass)	0.75 ± 0.27	0.65 ± 0.13	0.24 ± 0.46	0.81 ± 0.13
Cumulative N_2O emission ($mg\,kg^{-1}$ total biomass)	130.14 ± 18.91	156.92 ± 75.49	154.14 ± 75.74	122.28 ± 64.46
Total greenhouse gas emission in CO_2e ($g\,kg^{-1}$ plant total biomass)	206 ± 9.0 a	168 ± 52 ab	142 ± 49 ab	120 ± 29 b

Note: Data are presented as mean ± 1 × SE ($n = 4$). Numbers with the same letter are not significantly different at $p = 0.05$ and no letter following the numbers indicate no significant differences among treatments at $p = 0.05$.

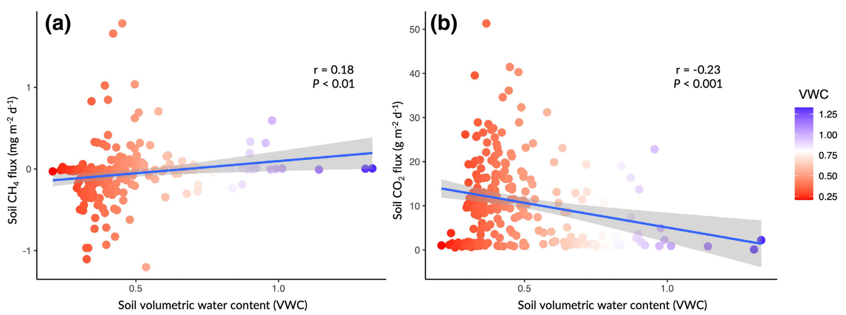

FIGURE 4 Correlations (Pearson's r, p-value) between soil volumetric water content and (a) soil CH_4 flux or (b) soil CO_2 flux over one growing season (October 2021–March 2022) in a field experiment in Madera, California, USA.

DISCUSSION

Soil physicochemical responses and nutrient leaching below the rooting zone

Biochar and biochar co-compost significantly increased surface soil total C by 24%–30% compared to control and compost (Table 1), suggesting that biochar was able to improve short-term soil C storage (Cooper et al., 2020; Gao et al., 2017; Majumder et al., 2019). All three organic amendments significantly increased soil water holding capacity, infiltration rate, microbial biomass, and soil net N mineralization compared to the control over one growing season (Table 1). The manure compost and the biochar co-compost used in our study were rich in N (i.e., having a C:N ratio of 18–24, Table S2). The N input from compost and biochar co-compost (i.e., 430 $kg\,ha^{-1}$ N with compost application and 340 $kg\,ha^{-1}$ N with co-compost application) likely stimulated soil net N mineralization and increased microbial C demand (Bengtsson et al., 2003; Chen et al., 2022), and soils amended with compost and co-compost were found to have a C:N ratio of 13 at the end-growing season (Table 1). The biochar used in our study had a low N content; it is, therefore, possible that the positive effect of biochar on soil N mineralization (and microbial biomass) was mainly driven by changes in abiotic factors, such as an increase in soil water holding capacity that would have indirectly enhanced microbial

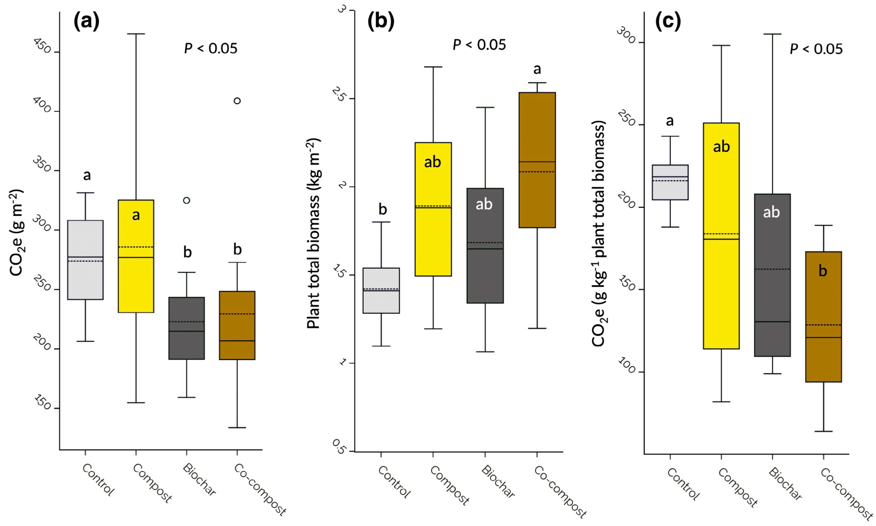

FIGURE 5 Influence of compost, biochar, and biochar co-compost on (a) soil total greenhouse gas emission in CO_2e ($g\,m^{-2}$), (b) plant total biomass ($kg\,m^{-2}$), and (c) soil total greenhouse gas emission in g CO_2e kg^{-1} plant total biomass over one growing season (October 2021–March 2022) in a field experiment in Madera, California, USA. Data were compared using Tukey-HSD test following ANOVA. Solid line indicates median, dashed line indicates mean. Boxes with the same letter annotated are not significantly different at $p = 0.05$.

TABLE 3 Model statistics for plant total biomass regressed against organic amendment treatments and soil metrics at the end-growing season (March 2022) in a mixed linear model (model fit $R^2 = 0.94$, adjusted $R^2 = 0.91$, $p < 0.001$).

Coefficients:	Estimate	SE	*t*-value	Level of significance
Intercept	−7.24	2.58	−2.80	*
Compost	0.54	0.06	9.01	***
Biochar	0.22	0.10	2.19	*
Co-compost	0.60	0.10	5.95	***
pH	1.38	0.26	5.31	***
Mean weight diameter	−0.19	0.10	−1.87	ns
Net N mineralization	−0.34	0.12	−2.72	*
NO_3^-–N	−3.64	0.83	−4.38	***
Microbial biomass C	−0.31	0.09	−3.40	**
Soluble inorganic P	0.52	0.07	7.31	***
Active inorganic P	0.43	0.21	2.00	ns
Labile organic P	−0.65	0.13	−4.99	***
Moderately stable inorganic P	1.87	1.12	1.67	ns

Note: All data were log transformed in the model to ensure data normality and homogeneity of variance.
Significance levels: *$p < 0.05$, **$p < 0.01$, ***$p < 0.001$, ns indicates $p > 0.1$.

activity and N mineralization (Gao et al., 2016; Gao & DeLuca, 2020; Jones et al., 2011).

Biochar and biochar co-compost significantly increased surface soil nitrification potential (Table 1). It is important to note that the soils in our field experiment were alkaline and had a pH of 8.1, but biochar and biochar co-compost both reduced soil alkalinity and decreased soil pH by 0.3–0.4 units (Table 1). It is possible that the functional acidic groups on biochar surface directly reduced the alkalinity of the soils over short term (Tomczyk et al., 2020). Alternatively, it is possible that the decrease in soil pH in response to biochar addition was driven by an accelerated soil nitrification which was commonly associated with H^+ production. Numerous studies have shown that the presence of biochar can increase soil autotrophic nitrification rate under a variety of mechanisms, for example, the high porosity of biochar providing more oxygenated microsites stimulating the activity of nitrifying microbial community (Liu et al., 2012; Wu et al., 2014); biochar adsorbing organic compounds that would otherwise inhibit nitrification (Ball et al., 2010; DeLuca et al., 2006). Compared to biochar, the additional N from biochar co-compost might have provided more NH_4^+ as a substrate for nitrification; and the higher infiltration rate in soils with biochar co-compost might have resulted in more oxygenated microsites, further stimulating nitrification and subsequently contributing of a decrease in soil pH (Gao & DeLuca, 2020, 2022).

For soils amended with biochar alone, the higher soil N mineralization and nitrification potential (compared to control) did not lead to a relatively higher soil NH_4^+-N or NO_3^--N content in surface soils throughout the season (Figure 1). It is likely that the mineralized NH_4^+-N and NO_3^--N were immediately taken up by microbes in surface soils where the C-rich, N-poor biochar might have resulted in N-limitation in soil microsites and increased microbial N demand in the 'charosphere' (Gao et al., 2019; Gao & DeLuca, 2021). In the biochar plots, the lower inorganic N content in surface soils and the higher water holding capacity might explain the lower N leaching compared to the control (Figure 1). In contrast, soils with biochar co-compost application had higher surface NH_4^+-N and NO_3^--N content, but tended to exhibit the least NH_4^+-N and NO_3^--N leaching loss below the rooting zone compared to biochar or compost amendments (Figures 1 and 2). The higher water holding capacity in soils with co-compost (Table 1) may explain the soil N retention pattern with co-compost throughout the season (Agegnehu et al., 2016b; Agegnehu et al., 2016; Razzaghi et al., 2020). Alternatively, the inorganic N at the rooting depth might have been actively taken up by winter wheat plant roots toward the end-growing season eliminating its leaching loss; and the N retention pattern was more pronounced in co-compost plots where we observed the highest plant total biomass among all treatment (Table 2). It is important to note that the application of compost had a higher total N input than co-compost in our experiment, yet biochar co-compost resulted in higher surface soil N content, less N leaching, and higher plant biomass. This finding suggests that biochar, as a composting media, has the capacity to significantly improve ecosystem N retention in a conventional wheat cropping system over short term.

Compost and biochar co-compost used in our study had similar total P content (2.2 $g\,kg^{-1}$), yet soil P availability only responded to biochar co-compost application (Table 1). Direct sorption of manure compost P onto biochar surfaces and the subsequent P retention over the growing season might partially explain the positive soil P response (Gao & DeLuca, 2020). It is also possible that the presence of biochar influenced microbial community composition that were responsible for organic P mineralization or phosphatase activity (Gao & DeLuca, 2018; Tian et al., 2021; Yang & Lu, 2022).

Soil CO_2, CH_4, N_2O, and NO_x emissions

We found that cumulative CO_2 emission from compost application was higher compared to biochar application over the entire growing season (Figure 3a and Table 2). While biochar was C-rich and N-poor (C:N ratio of 411), compost was N-rich (C:N ratio of 18) and the C contained in the compost was more biodegradable compared to the C contained in biochar. It is therefore likely that the difference in CO_2 emission was mostly driven by active microbial metabolic processes decomposing the compost material under sufficient N supply (Ding et al., 2010; Gross et al., 2022; Iovieno et al., 2009). Given that there was minimal resource competition between microbes and wheat plants at the early season, it is not surprising that most of the CO_2 emission and treatment effect were observed at the first month of the field experiment (Figure 3a and Figure S2a). Compared to compost, biochar co-compost used in our study had a slightly higher C:N ratio (24:1) and resulted in a relatively lower cumulative soil CO_2 emission over the season (Figure 3a and Table 2). The finding of the highest microbial biomass and the lowest cumulative CO_2 emission associated with the use of co-compost among all organic amendments implies a high microbial C use efficiency, which may subsequently indicate a high potential to increase soil C storage and persistence through the contribution of microbial necromass (Kästner et al., 2021; Liang et al., 2020; Zhang et al., 2021).

While no significant treatment effect was detected for the cumulative CH_4 emission, we found that soils amended with biochar tended to exhibit lower CH_4 uptake than compost or biochar co-compost at the early and mid-growing season (Figure 3b and Figure S2b). Surface soil volumetric water content in biochar plots was consistently higher than that in compost or biochar co-compost plots over the course of the field trial (Figure S3). It is likely that biochar resulted in water saturation in soils, reduced the oxygen availability and increased the number of anaerobic soil microsites (van Zwieten et al., 2015; Yu et al., 2013), which subsequently led to higher CH_4 fluxes with lower CO_2 fluxes (Figure 4 and Table S3).

It is surprising that no treatment effect was found in cumulative soil N_2O, NO, or NO_x emission over the growing season (Table 2 and Figure 3c) considering that compost and biochar co-compost had 430 $kg\,ha^{-1}$ and 340 $kg\,ha^{-1}$ N input compared to 20 $kg\,ha^{-1}$ N input in the biochar treatment and no additional N input in the control. Instead, soil N_2O, NO, and NO_x emissions seemed to be more responsive to the season than the treatment where relatively higher emissions occurred in the first month of the field trial when temperature was higher and residual N from previous cropping cycles was likely more available (Figure 3c and Figure S2c). It is important to note that the field was not fertilized during the winter and the majority of the N supply at the early winter season was from residual N from synthetic fertilizers applied in the previous cropping cycle as well as the mineralization of organic residues left from the previous summer growing season (e.g., tomato plants). With limited plant N uptake and minimal plant–microbial N competition at the early season, N might have been easily lost through NO_3^- leaching or gaseous loss, especially through ammonia volatilization in the high pH soils.

It is important to note that all organic amendments were applied at equivalent C rates (8 Mg C ha^{-1}), yet biochar and biochar co-compost both resulted in a significantly lower cumulative CO_2 emission without increasing soil CH_4 or N_2O emission over one growing season (kg^{-1} C input), and biochar co-compost resulted in the lowest total greenhouse gas emission (g CO_2e kg^{-1} plant total biomass). This finding indicates that biochar co-compost has the highest potential among all treatments in climate change mitigation while being a beneficial amendment for soil C sequestration and agroecosystem productivity. This finding adds to previous work showing the substantial climate change benefit achieved during the production of biochar co-composting. The emissions associated with producing the compost and biochar co-compost used in this study are reported in Harrison et al. (2022), which found a 84% reduction in methane emissions when dairy manure was composted with 6% (w/w) biochar relative to compost without biochar. Now, we show that the climate benefit of biochar co-composting is not limited to just the composting process, but that biochar co-compost application to soils can also mitigate climate change by enhancing an ecosystems capacity to act as a greenhouse gas sink.

Plant performance

We found that higher soil N and P availability significantly explained changes in plant total biomass, where the biochar co-compost application resulted in the highest plant total biomass over one growing season among all treatments. This finding suggests that soil nutrient availability was likely a key limiting factor regulating plant performance or productivity during the winter season when the farmers did not implement any additional fertilization. This finding is consistent with many other studies where the authors reported an approximately 30%–40% increase in wheat yield following the application of biochar compost mixtures or biochar co-compost (Antonangelo et al., 2021; Wang et al., 2019). For instance, Lashari et al. (2013) found that the significant increase in the wheat yield following biochar co-compost application over one season was largely driven by an improved soil nutrient availability and the salt stress amelioration effect of their biochar co-compost (Lashari et al., 2013). Similarly, Qayyum et al. (2017) found that the increases in soil N, P, and K availability were effectively reflected in the tissue nutrient concentrations of wheat crops (Qayyum et al., 2017). In our study, over one growing season, winter wheat grown in soils amended with compost and biochar co-compost also had higher tissue concentrations of total P, Ca, and Fe compared to those in soils amended with control or biochar. This finding suggests that the nutrients in dairy manure are likely to be retained during the composting process and the presence of biochar possibly have further improved the nutrient retention via various mechanisms (Steiner et al., 2015). Higher soil P availability in soils amended with co-compost was reflected in plant P concentration, which was found to explain a significant variation of the total plant biomass. Given that the silage is commonly used as dairy feed on-farm, the findings in our study partially indicate that the biochar manure co-composting and subsequent soil application could be an efficient way to recycle P from the feedstock materials, retain P in the system, and close the nutrient loop.

CONCLUSION

We show that the application of all three organic amendment treatments significantly increased total winter wheat biomass compared to the control, and that biochar co-compost resulted in the lowest nutrient leaching loss potential relative to the application of biochar or compost on their own over the course of one growing season. Biochar co-compost likely improved plant growth by significantly increasing water holding capacity and by reducing the leaching of NH_4^+ and ortho-P. Both biochar co-compost and biochar treatments significantly reduced soil cumulative GHG relative to the compost treatment or no-amendment control, suggesting that, in terms of climate mitigation, biochar co-compost performs similar to biochar, which has a well-documented high potential to mitigate soil GHG emissions (van Zwieten et al., 2015). Our study suggests that biochar co-compost has advantages beyond the composting process, and its production and application offers a climate-smart opportunity to close nutrient loops between livestock and cropping systems while maximizing benefits to farmers.

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CONFLICT OF INTEREST

The authors declare no conflicts of interest.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study is available in Dryad: https://doi.org/10.6071/M3MX1B.

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